Does the Lack of Reference Ecosystems Limit Our Science? A Case Study in Nonnative Invasive Plants as Forest Fuels

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ABSTRACT

In forest experiments the problem of inadequate controls often arises. True controls might not be required in case studies, comparisons along an environmental gradient, or comparisons of multiple treated and untreated areas. In a recent characterization of fuels in invaded and uninvaded forest conditions for four forest types at 12 locations in Maine, Maryland, Massachusetts, New Jersey, New York, Vermont, and Virginia, high-quality reference stands usually were not available as true controls. We called the uninvaded areas "comparison areas," and applied a modified planar sampling technique to quantify live and dead fuels. No overarching pattern emerged; fuels in fire-adapted pitch pine differed from the three other forest types in that stands invaded by black locust had fewer 1- and 10-hour fuels, but more forbs cover and higher basal area. Invasive shrubs increased fuel height and density across most forest types. Invasive grasses in forests present an underrecognized hazard fuel if drought ensues. The comparison stand study design enabled uncovering of significant differences between invaded and uninvaded stands, especially in hardwoods and mixed woods, and fuels in softwoods were less affected by invasive plants.

Keywords: control, reference conditions, wildland urban interface, fire, fuels, invasive plants

Effective forest research depends greatly on our ability to assess whether management activities produce reliable results. Where there is no benchmark against which to compare a treatment, how do we know when our objective has been reached? This was the subject of a technical session during the 2003 Society of American Foresters National Convention in Buffalo, New York, which was organized by the C1 Forest and Range Ecology Working Group. Including this article, six of the presentations from that session are reported in this issue (Asbjornsen et al. 2005, Frelich et al. 2005, Goebel et al. 2005, Kenefic et al. 2005, and Stephens and Fulé 2005).

Regardless of the nature of a research question, in an experiment, the control is that area (or group of individuals) that receives no treatment (Ford 2000). From a statistician's viewpoint, the control presents a set of random variants about a central tendency, with no identifiable force at work on a process. The area that receives treatment is identical to the control except for the treatment itself. Achievement of true control conditions typically is accomplished in a laboratory setting; field experiments often must allow for variation among control units and provide sufficient sample size that this variability does not override the effects of factors to be tested.

In field studies of forest ecology, a control implies an unmanipulated area that can be compared with the treated area. Sometimes it is convenient for researchers to ignore the fact that a forest stand is an open system, not static but continuing to be influenced by past fires, storms, pests and diseases, animal population dynamics, pollution, invading plants, and prior human activities. In exemplary studies, researchers try to account for the many interacting variables that are part of a stand's ecology and history and distinguish treatment response in light of those features.

Much as we might wish to compare today's Northeastern forests to a "forest primeval"—whatever that may be—we can not fully *know* the conditions before major deforestation by European-American settlers during the 17th through 19th centuries and we do not fully understand the extent to which Native Americans used forest resources, altered the fire regime, and otherwise influenced forest vegetation since the Wisconsin glaciation receded (Pielou 1991), particularly from ca. 9,000 years ago. The utility of estimated pre-European settlement conditions is exemplified in the ongoing development of a federally funded ($40 million from FY04 to FY09) nationwide mapping project, LANDFIRE, that allows assessment of fire regime condition class and seeks to reveal departure of current vegetation, fuels, and disturbance regimes from historical reference conditions. Reduction of hazard fuels and ecological restoration can be improved by comparing pre-1600 vegetation conditions and historic fire regimes to models of potential natural vegetation, but these are estimates only, based on expert opinion, and the expectation of "proof" is untenable.

As an example of our inability to fully define reference conditions for forests, consider pollen stratigraphic records from lake sediments. These use pollen exines and macrofossils to provide detailed information on the change in distribution of wind-pollinated vegetation in response to climate change (e.g., DeHayes et al. 2000). Despite the reliability of such information, there are few data in the pollen record regarding animal-pollinated plants. As a result, past forest composition must be reconstructed with only broad approximations about many understory plants and insect-pollinated trees, including the gums (*Nyssa* spp.). These animal-pollinated components include many fleshy-fruited plants that support forest birds and other wildlife (e.g., low sweet blueberry, *Vaccinium angustifolium* Ait.), and the lack of information about their response to climate change is a major gap.

Despite limitations, there is value in creating approximations of old-forest conditions that provide unusual habitat (Frelich et al. 2005) because biodiversity depends on every successional stage. Before effective implementation of restoration activities can be undertaken, characterization of the reference system is crucial (Asbjornsen et al. 2005, Goebel et al. 2005). Definition of reference information allowed Asbjornsen and coworkers (2005) to quantitatively assess restoration outcomes in a midwestern oak savanna system. Through this approach, they identified restoration goals that are quickly achieved and those that will require more extensive efforts, thus focusing on future management. Additionally, they explored the development of reference information through experimentation and adaptive management.

Stand development has been influenced for hundreds of years by climate and perhaps by fire (Stephens and Fulé 2005), wind, and ice. With respect to human-induced disturbances at the regional scale, often, we must rely on historical accounts such as lumber tallies, which can be rife with losses and inefficiencies that lead to significant underestimates of, e.g., eastern white pine (Wilson 2005). To refine forest management strategies, Betts and Loo (2002) compared efficacy of the "witness tree" method and a "potential forests" approach for characterizing pre-European settlement conditions in forests of New Brunswick. They recommended combining the two approaches. Other types of extrapolations are available for many regions and forest types (e.g., Benson 1964, Cronon 1983, Lindholdt 1988, Lawrence 1991, Williams 1992, Whitney 1994).

Comparisons between treated areas and untreated "controls" on a given site can take into account previous forest conditions based on the history of agriculture and harvest and will include current tree size, density and age, presence of stumps, stone walls, cellar holes, old fencing, historic photos, or other evidence. In addition, climatological changes, fire, pest and disease, ice storm, and catastrophic windthrow should be considered. Unless such aspects of disturbance are accommodated when the study is set up, the ability to generalize results will be limited.

The validity (do the data reflect what was supposedly measured?) and reliability (reproducibility) of some forest research could be at stake, depending on the research questions to be answered. In quest of a good control in forest experiments, researchers seeking to promote rigor in their experiments will want to review the ideas of wildlife biologists such as Keppie (1990), who discounts some of the long-held tenets of ecological study design, or Anderson et al. (2000), who have reassessed the testing of the null hypothesis in ecological studies. Green's (1979) 10 principles of good study design include the following: "To test whether a condition has an effect, collect samples both where the condition is present and where the condition is absent but *all else is the same*. An effect can only be demonstrated by comparison with a control" (italics added). In forest research for the Northeast, and probably for most parts of the United States, it could be difficult to support the premise that a silvicultural treatment can be compared, in a reasonably sized block, to a bona fide control that represents a completely unmanipulated stand. In heterogeneous forest ecosystems, a good control is virtually unattainable because of the complexity of environmental interactions. The control would have to be consistent with the treated area in forest-cover type (dominant and codominant trees in similar proportions), understory plant communities, soils, slope, and aspect. The two areas would have the same disturbance history, including fire, windthrow, ice storm, and human activities (agriculture and timber harvesting).

Depending on the research question that is being addressed, four or more experimental units per treatment level usually are expected, and the control is considered as a treatment level. This way, the variation within a treatment can be considered as well as any difference between treatments. The value of an experiment can be increased by adding outlying experimental units so that the geographical application of the study is extended, but this is not always feasible. In the best of all worlds, experimental units should be assigned at random across site conditions that are relatively homogeneous, and sufficient sample size will improve validity. This might mean that within each treatment level, there are more than 10, perhaps even 20, observations per attribute to be measured. In any case, limitations in the control must be recognized so that the results can be interpreted accurately. If the control does not match the treated area in the criteria mentioned or if the study design was inherited and is deficient in other ways, there still are some compelling ways to improve the reliability and validity of the study.

Controls as such are not part of the study design used by the Forest Inventory and Analysis (FIA) Program in the USDA Forest Service. Their widely separated samples, with plots on a grid of 5-km (3.11 mi) intervals, are the backbone of data sets that have rich potential to answer landscape-scale questions on ecological forestry, land use, and climate change. Recently, Chojnacky et al. (2004) analyzed FIA data on 778 plots from several eastern states to develop regression models that predict biomass of down woody debris and shrub/herb cover, among other features.

In another example, controls might not be needed if the data can be presented as a case study or when the results are unequivocal. Well-established long-term studies such as that in the Penobscot Experimental Forest in the Acadian spruce-fir forest of Maine have become increasingly valuable with age and repeated measurements, often leading to new research avenues not conceived or explored by the original researchers. Despite their enduring value, such studies can present numerous challenges for interpreting results (Kenefic et al. 2005).

Ways to adjust for inadequate controls when setting up a study design include the following: (1) contrasting the treatment, which is replicated, with an equal number of


A B

Figure 1. (a) Stand dominated by native, fire-adapted pitch pine, maintained using prescribed fire at the Albany Pine Bush in Albany, New York. (b) On a former agricultural field, sandy soils can become invaded by less flammable black locust, which is understood to be outside its native range north of Pennsylvania.

untreated blocks that share the same cover type; the emphasis is on multiple replicates for all treatment levels—this makes possible an analysis of variance for estimating not only the difference between the treatment and the untreated areas (which are not actually a control), but also the within-treatment variation; (2) capturing a gradient of conditions and using environmental variables to explain species occurrences (e.g., Dibble et al. 1999); an ordination technique can pinpoint the environmental variable(s) that have the greatest influence on plant species diversity or occupation of habitat (McCune and Grace 2002); (3) measuring the response to the treatment in one or several species rather than with an overall feature such as total basal area for the stand, and ensuring that the species chosen are well represented in both the treated and the reference stands; and (4) calling the reference conditions a comparison rather than a control; in statistical terms, "control" has a specific meaning discussed previously. In eastern North America, if indeed no stand is free of effects because of peoples' activities (including acid deposition, increase in CO_2, and global warming; i.e., International Panel on Climate Change 2001), then *comparison* is a better word choice for the reference condition.

By uncoupling the concept of the comparison stand from the implication that there is an untreated control, the researcher may be able to show a response to the treatment without untangling effects of stand history and the many other influences on the current landscape. As in any science, with any approach, we want to avoid extending beyond the data. Interpretation of the findings must be restricted to the area(s) studied. We can not assume generalizability that the pattern applies over a region or forest type, when only a small portion of the system was measured. On the other hand, generalizability of the FIA data is presumed because of the scale and extent of the grid on which plots were established.

Using the Comparison Approach in a Fuel Characterization Study

We conducted a project in which we used comparison areas instead of controls. We collected data to test the hypothesis that invasive, nonnative plants alter fuels in eastern forests by either increasing or decreasing the fuel load. We could find few or no forest stands that had had no agricultural or timber harvest activity. Other influences were in evidence. For example, balsam woolly adelgid (*Adelges piceae* [Ratzeburg]), a nonnative insect pest, has caused mortality of balsam fir (*Abies balsamea* [L.] P. Mill.) at Holbrook Island Sanctuary in Brooksville, Maine.

We undertook this study because in our region, fuels in the wildland urban interface (WUI; the proximity of human populations to fuels that could be consumed by a wildfire) have not been measured extensively, and managers who model fire behavior using BEHAVE fuel models must estimate conditions based on data from the vegetation of other regions. In all states in our region, residences and businesses are near or within forests. In wetter years, fire is of little concern, although sporadic droughts have created conditions leading to catastrophic wildfires in the eastern United States. For example, in Oct. 1947 in Maine, according to newspaper accounts, nearly 86,235 ha (213,000 ac) burned; 35 towns were affected, 851 year-round residences were destroyed, and 397 seasonal cottages were lost. The largest single fire that year was in southern Maine and it consumed nearly 66,800 ha (165,000 ac), with estimates ranging from 60,730 to 72,875 ha (ca. 150,000 to 180,000 ac). The fire stopped only when it reached the Atlantic Ocean. Influence of this fire on the eastern white pine—northern red oak type—red maple type was estimated as part of a recent inventory of the 1,498-ha (3,700-ac) Massabesic Experimental Forest in southern Maine (Dibble et al. 2004).

Invasive plants can alter fuels by changing the (1) height and density of the fuel bed, (2) phenology of green-up and browning of vegetation, (3) flammability of the vegetation through arrangement of fuels in three-dimensional space or through volatile chemicals in plant parts, among other ways. Over time, invasive plants can alter forest composition by affecting forest regeneration and even the fire regime. Multiple effects are possible when invasive plants become part of a fuel bed, and we suspected that fire-adapted pitch pine (*Pinus rigida* P. Mill.) forests respond differently than do other forest types of the Northeast. Among the most prominent invasive plants of pitch pine forests in our region is black locust (*Robinia pseudoacacia* L.; Figure 1), which is native from Pennsylvania southward (Fernald 1950) but not in New York or New England. Some managers consider black locust trees to be "nonflammable." In pitch pine sites, black locust is expected to lengthen the fire-return interval to an unsustainable level in that pitch pine eventually becomes shaded out along with associated native fire-adapted understory plants. Although this decreases fire danger in the WUI, invasion by black locust affects plant biodiversity. It also destroys habitat for the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis* Nabakov) and its disturbance-dependent host plant, blue lupine (*Lupinus perennis* L.). This blue lupine is not the colorful Russell or bigleaf lupine of roadsides in coastal Maine, which is the introduced *Lupinus. polyphyllus* Lindl. of the Pacific Northwest. Russell lupine is considered a highly invasive pest in New Zealand and Finland.

Certainly not all forests in the Northeast are invaded, and invasive plants tend to be patchy where they are present; however, there are numerous invasive plant species that form dense, persisting populations in eastern forests (Richburg et al. 2001). Some were introduced as "conservation plantings" or garden subjects and then escaped from their original plantings or if fleshy-fruited, were spread from plantings into forests by birds. Some have spread in part because white-tailed deer do not favor them as food and because they are aggressive competitors for space, light, water, and nutrients. A shade-tolerant Eurasian grass, Japanese stiltgrass (*Microstegium vimineum* Trin. A. Camus), persists under a closed canopy on mesic soils and is an example of a plant that increases the load of fine fuels in the forest.

Invasive plants have many influences besides their potential alteration of fuel loads and fire regimes. They impact on biodiversity by occupying rare plant habitat; they degrade animal habitat in that they constitute a different food and cover resource than native plants with which such animals evolved over the millennia, and they can decrease the quality of a recreation experience, although visitors might not be conscious of the problem. This is especially so at a historic site where vegetation is intended to resemble the way the landscape looked on the eve of a Civil War battle (e.g., at Manassas National Battlefield Park, Manassas, Virginia).

In addition to black locust, invasive trees that are common in the Northeast are tree-of-heaven (*Ailanthus altissima* [P. Mill] Swingle), Norway maple (*Acer platanoides* L.), and apple (*Malus* sp.), although apple has been encouraged to attract wildlife. All these can spread and persist long after the forest canopy closes, and they might crowd out native trees over time. Invasive shrubs and vines that have been documented in Northeastern forests include Japanese barberry (*Berberis thunbergii* DC) and common barberry (*B. vulgaris* L.), Asian honeysuckle (e.g., *Lonicera* x*bella*, *L. tatarica* L., *L. xylosteum* L., *L. morrowii* Gray, and others), Japanese honeysuckle (*L. japonica* Thunb.), privet (*Ligustrum* spp.), alder-buckthorn (*Frangula alnus* P. Mill.), common buckthorn (*Rhamnus cathartica* L.), Asian bittersweet (*Celastrus orbiculata* Thunb.), and multiflora rose (*Rosa multiflora* Thunb. *ex* Murr.). Forbs that affect fuel beds include garlic mustard (*Alliaria petiolata* [Bieb.]) Cavara&Grande). Nonnative grasses can be present in forests despite a closed canopy, and these include Japanese stiltgrass (*M. vimineum*), wood bluegrass (*Poa nemoralis* L.), fine-leaved sheep fescue (*Festuca filiformis* Pourret), and sweet vernal grass (*Anthoxanthum odoratum* L.). We informally observed that most of these plants tolerate shade, which allows them to invade a forested environment whether or not it has been disturbed recently. Some woody species grow faster and develop a larger crown in openings (e.g., Asian bittersweet, multiflora rose, barberry, and apple), and respond rapidly after disturbance to a forest stand.

Methods

From 2000 to 2003, we quantified fuels in forests with nonnative, invasive plants (Dibble et al. 2003a, 2003b). At 12 sites (including 13 study areas) on mostly federal lands, with some state and private, nonprofit lands in Maine, Maryland, Massachusetts, New Jersey, New York, Vermont, and Virginia (Table 1), representing four broadly defined forest types, we sampled forests with and mostly without invasive plants. We used a modification of the widely used planar intersect technique (Brown 1974, Brown et al. 1982) to sample fuels, and sampled live vegetation on fixed and variable plots.

The four forest types were hardwoods (>50% hardwood tree species composition), mixed woods (softwood species, 26–50%), softwoods (>50% softwood tree species), and pitch pine (>80% pitch pine). Percentages were for uninvaded conditions.

Sampling Layout. In consultation with the land manager for each site, we surveyed for comparison areas. We sought areas in which a single transect or series of transects would span patches of both heavily invaded and uninvaded forest. Where this was not possible, we chose pairs of proximal stands; the maximum distance between areas was ca. 4 km (2.5 mi). We sought similarity in overstory, disturbance history, soils,

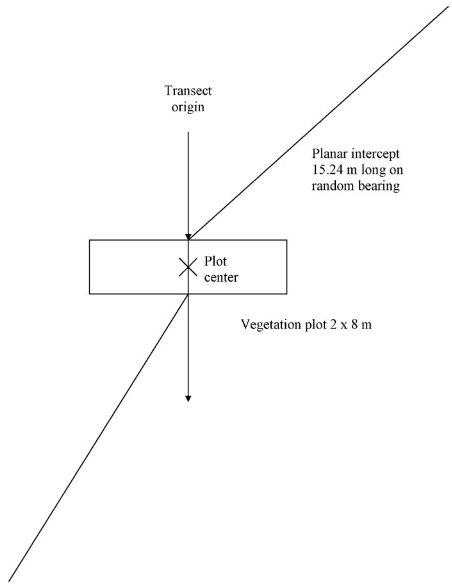

Figure 2. Plot layout and planar intercept orientation at a random point along a transect of at least 100 m (328 ft) and >5 m (typically <10 m) from next nearest plot (not to scale).

slope, and aspect. Areas completely free of any nonnative plants were not always available for study, and in some locales a low density of woody invasive species were present in conditions that we called "slightly invaded" and used to compare with densely invaded stands. We expected that multiple study areas per forest type would improve validity and that data interpretation would have to rely on significant differences between invaded and uninvaded conditions. We collected species abundance data along transects 4 m (13.12 ft) wide—in some cases these were parallel—and >30 m (98.4 ft) apart and >50 m (164 ft) long, running perpendicular to a road, trail, or other ecotone. The study area was at least 400 m (1,312 ft) square for each condition.

Fuel and Vegetation Sampling. At each study area and within each condition we established plot centers on the transects at five random locations at least 5 m (16.4 ft) and usually at least 10 m (32.8 ft) apart. We georeferenced plots and noted their aspect, slope, and obvious disturbance history based on features such as stone walls, charcoal, or cut stumps. From 1 m (3.28 ft) on each end of a plot center (Figure 2), we measured fuels on 2- to 4-planar intercepts established on random bearings, using a modification of the method developed by Brown (1974) and Brown et al. (1982) and refined in consultation with William A. Patterson III of University of Massachusetts, Amherst, and

Table 1. Features of 13 study areas at 12 sites, including location, ownership, vegetation aspects, broad forest–type category, number of plots, and site characteristics.

Site (abbreviation)	Location	Ownership	Total ha at the site	Name of study area	Cover	Forest type designation	Nonnative invasive spp.	Transect length (m)	No. plots	Aspect	Slope (°)
Acadia National Park (AC)	Bar Harbor, Hancock Co., ME	National Park Service	14,165	Great Meadow (i) The Tarn (u)	Poplar species (i) Red oak/red spruce (u)	Hardwoods	Eastern ninebark, Shrubby St. Johnswort, fine-leaved sheep fescue	300 (i) 300 (u)	10	W (i) W (u)	0–10 (i) 0–20 (u)
Albany Pine Bush Preserve (AF)	Albany, Albany Co., NY	Albany Pine Bush Preserve Commission	809	Firebrand (i) Friendly (u)	Black locust (i) (u)	Pitch pine	Black locust	Firebrand = 100 (i) Friendly = 235 (u)	10	Variable for both i and u	10–30 (i) 0–20 (u)
Albany Pine Bush Preserve (AL)	Albany, Albany Co., NY	Albany Pine Bush Preserve Commission	809	Locust (i) Chubb (u)	Black locust (i) (u)	Pitch pine	Black locust	Locust = 227 (i) Chubb = 200 (u)	10	W (i) N (u)	0 (i) 0–17 (u)
Antietam National Battlefield (AN)	Antietam, Washington Co., MD	National Park Service	394	Snavely Ford Woods	Oak/hickory hardwoods	Hardwoods	Japanese honeysuckle	184 (i) 142 (u)	10	S (i) E (u)	0–20 (i) 20–30 (u)
Cape Cod National Seashore (CC)	Wellfleet, Barnstable Co., MA	National Park Service	17,647	Fresh Brook	Black locust (i) Pitch pine (u)	Pitch pine	Black locust, sweet vernal grass	200 (i) 200 (u)	10	S (i, u)	3–14 (i) 2–50 (u)
Finger Lakes National Forest (FL)	Hector, Schuyler Co., NY	USDA Forest Service, National Forest System	6,221	Mark Smith Rd	Oak/hickory hardwoods	Mixed woods	Multiflora rose		10	SE (i) S (u)	3–12 (i) 5–20 (u)
Holbrook Island Sanctuary (HI)	Brooksville, Hancock Co., ME	State of Maine Department of Conservation	498	Hutchins Estate	Red spruce/Balsam fir	Softwoods	Norway maple, wood bluegrass	150 (i) 70 (u)	10	WNW (i) S (u)	0–5 (i) 10–25 (u)
Manassas National Battlefield Park (MA)	Manassas, Prince William Co., VA	National Park Service	2,024	Brawner Woods (i) Carter Woods (u)	Oak/hickory hardwoods	Mixed woods	Japanese honeysuckle	158 (i) 260 (u)	10	WNW (i, u)	5–10 (i, u)
Merck Forest and Farmland Center (MK)	Rupert, Bennington Co., VT	private, nonprofit	1,275	Stone Lot	Mixed hardwoods	Hardwoods	Asian honeysuckle	94.5 (i) 102.5 (u)	10	S (i) SE (u)	15–28 (i) 15–30 (u)
Massabesic Experimental Forest (ME)	Lyman, York Co., ME	USDA Forest Service, Northeastern Research Station	1,488	Administrative Unit	Red oak–white pine	Mixed woods	Oriental bittersweet	144 (i) 70 (u)	10	WSW (i, u)	2–10 (i, u)
Morristown National Historical Park (MO)	Morristown, Morris Co., NJ	National Park Service	686	Jockey Hollow	Oak–yellow poplar hardwoods	Hardwoods	Japanese barberry	150 (i) 150 (u)	10	NE (i, u)	3–15 (i) 5–10 (u)
Penobscot Experimental Forest (PE)	Bradley, Penobscot Co., ME	University of Maine	1,538	Old Orchard	Oak–big-tooth aspen	Hardwoods	Alder-leaved buckthorn	96 (i) 100 (u)	10	W (i, u)	0–5 (i) 0–5 (u)
Rachel Carson National Wildlife Refuge (RC)	Kittery, York Co., ME	Fish and Wildlife Service	1,902	Brave Boat Harbor Unit	Eastern white pine on old fields and mixed hardwoods	Softwoods	Japanese barberry	150 (i) 151 (u)	10	NE (i) various (u)	0–5 (i) 0–16 (u)

i, Invaded conditions; u, uninvaded conditions. Aspect is abbreviated cardinal directions (e.g., N, North; NE, Northeast).

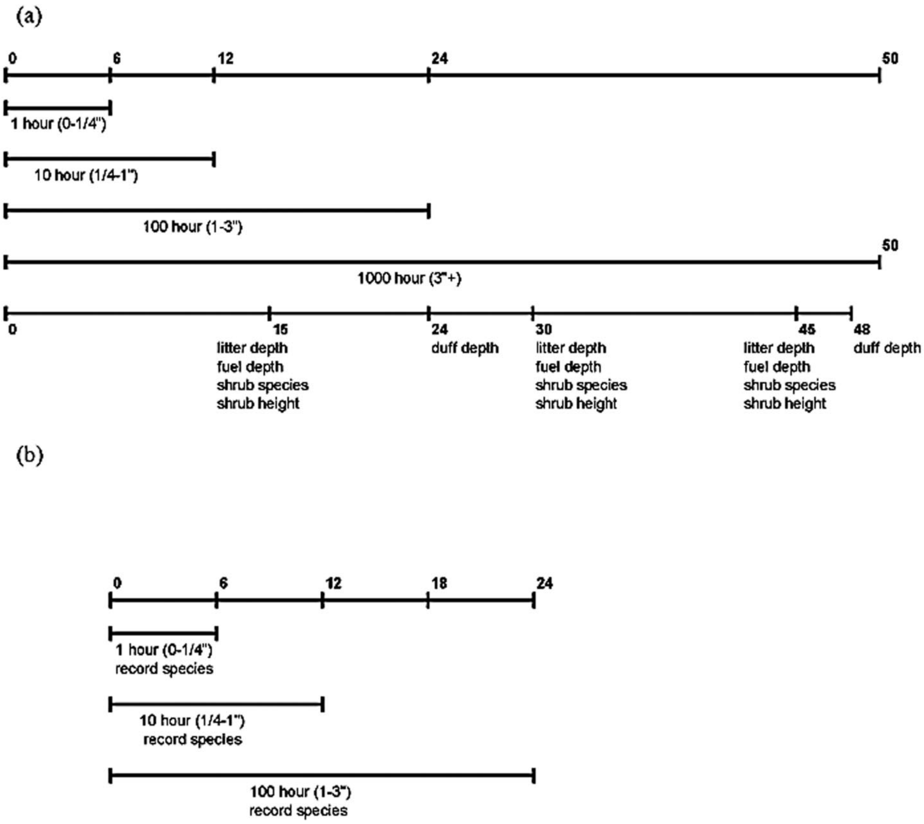

Figure 3. Length of sample segments and points for sampling various fuel depths along (a) a modified planar intercept for sampling fuels at 13 study areas in 12 locales from 2001 to 2002 and (b) a partial planar intercept used in 2003 to obtain additional hour fuels data. Units are in feet (and inches) after Brown (1974).

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On each planar intercept we recorded slope along the bearing and tallied the number of dead, detached, woody fuels in each diameter size category (Brown 1974; 1 hour, ≤0.64 cm [¼ in.]; 10 hour, >0.64–2.54 cm [to ≤1 in.]; 100 hour, >2.54–7.62 cm [1 to ≤3 in.]; and 1,000 hour, >7.62 cm [3 in.]; litter and duff depth, fuel depth; and shrub height (by species) along designated portions or points of the intercept (Figure 3). If the fuel was in direct contact with the litter where it intersected the line, we tallied it as "within" the fuel bed, otherwise it was "above" the fuel bed. For data analysis, we combined within and above fuels.

Sampling intensity for hour fuels was from 10 full planar intercepts and 10 partial intercepts from most of the 13 study areas. Samples were segments that totaled at least 36.59 m (120 ft) of 1-hour fuels data, 73.17 m (240 ft) of 10-hour fuels data, and 146.34 m (480 ft) of 100-hour fuels data per condition in each of the 13 study areas. Averages were used in data summaries for fuel attributes, based on 10–21 samples (segments) per condition for 1-, 10-, and 100-hour fuels (average, 18.3 samples, 19.0 and 19.0 respectively, ±SD 3.65, 2.85, and 2.85). There were 10–15 samples (segments) for 1,000-hour fuels per condition (average, 9.81 ± 2.23). In addition, by condition and study area we took at least 27 measurements of fuel depth, litter depth, and shrub height, and at least 18 measurements of duff depth.

We clipped samples of live fuel to estimate biomass at a subset of sites (AC, CC, FL, HI, MO, and PE, site abbreviations given in Table 1; four to six samples per condition at each study area) by bagging all live vegetation in an area 0.305 m (1 ft) wide by 1.83 m (6 ft) long and 1.22 m (4 ft) high at predetermined random points on the planar intercepts. This material was dried to constant weight in a drying room at the University of Maine, Orono, sorted into nonwoody and hour-class sizes, and weighed. At sites where grasses were significant (AC, FL, CC, and HI; Table 1) we further sorted the nonwoody material into graminoids, forbs, shrubs, or bryophytes and weighed these individually (results to be presented elsewhere).

Sampling of live fuel and slash included ocular estimations of percent cover by stratum from plot center for a 3.05-m (10 ft) radius circular plot. Cover was assigned for forbs, graminoids, low shrubs, high shrubs, trees, and slash according to the following cover classes by percent: <1, 1–5, 6–25, 26–50, 51–75, and 76–100. Also, from plot center, we used a 10-basal area factor prism to sample trees by species, alive or dead. Alternating borderline trees were counted as "in." Canopy closure was estimated using a convex spherical densiometer at plot center. Biomass of nonwoody fuels was estimated from a 40 × 40-cm (15.75 in.) square plot taken 1 m (3.28 ft) from the plot center; nonwoody materials were bagged including the entire duff layer, which is down to but not including fungal hyphae, and summarized in metric tons/hectare.

For the purpose of assessing invasive plant density and influence of invasive species on native plant communities and tree regeneration, additional vegetation sampling was on 2 × 8-m (6.56 × 26.25 ft) plots (Figure 3). We assigned cover for each vascular plant species in five classes for cover <5%: rare (<5 individuals, count), infrequent (one or a few clumps of 5–10 individuals); occasional (numerous individuals, not common; must look around to find it), common (occurs ± everywhere), and overhanging (tree stem outside plot, foliage overhangs plot). We used five additional cover classes: 5–25%, 26–50%, 51–75%, 76–100% (visualized as the midpoint of each), and vicinity (not in plot but nearby, within about 20 m [65.6 ft] of plot). Voucher specimens where collected for species not readily identified in the field. To assess regeneration, we censused three randomly selected 1-m (3.28 ft) square subplots of the 2 × 8 m (6.56 × 26.25 ft) vegetation plot and counted seedlings, alive and dead, of woody plants <0.5 m (1.64 ft) tall. These results will be reported elsewhere.

Data Analysis Techniques. We used Brown's (1974) methods for deriving fuel loads from the planar intercept data, with some elaborations and modifications. To obtain the constant d^2, we measured the diameter of 1- and 10-hour fuels of identifiable species in two 40 × 40-cm (15.75 in.) biomass plots per condition. Estimates for 100-hour fuels are considered rough. Values for specific gravity (*s*) were from Forest Products Laboratory (1955) for specific

Table 2. Results of Wilcoxon two-sample tests on fuel attributes by condition as invaded or uninvaded for four broad forest types.

Variable (units)	Sum of scores, invaded (25 plots)	Sum of scores, uninvaded (25 plots)	One-sided probability < Z	Chi-square approximation, with 1 df
Hardwoods				
Nonwoody litter (kg/ha)	483.0	792.0	0.0014	8.986
100-hr fuels (kg/ha)	758.0	517.0	0.0099	5.468
Duff depth (cm)	512.0	763.0	0.0073	6.006
Graminoid cover (%)	730.5	544.50	0.0317	3.485
Shrub cover (%)	846.0	429.0	<0.0001	16.548
Tree cover (%)	575.0	700.0	0.0227	4.070
Basal area (m^3/ha)	542	733	0.0320	3.466
Shrub height (m)	734	541	0.0311	3.515
Shrub frequency	830.5	444.5	<0.0001	14.211
Mixed woods				
Duff depth (cm)	155.0	310.0	0.0007	10.452
Fuel depth (cm)	166.0	299.0	0.0031	7.615
Shrub cover (%)	307.5	157.5	0.0009	9.786
Slash cover (%)	290.0	175.0	0.0047	6.860
Tree cover (%)	202.0	263.0	0.0368	3.308
Shrub frequency	291.0	174.0	0.0077	5.97
Pitch pine				
Nonwoody litter (kg/ha)	168.0	297.0	0.004	7.157
1-hr fuels (kg/ha)	159.0	306.0	0.001	9.296
10-hr fuels (kg/ha)	191.0	274.0	0.045	2.963
Forbs cover (%)	272.5	192.5	0.043	3.033
Slash cover (%)	181.5	283.5	0.013	5.078
Basal area (m^3/ha)	274.0	191.0	0.044	2.988
Shrub height (cm)	288.5	176.5	0.011	5.398
Softwoods				
Nonwoody litter (kg/ha)	128.0	82.0	0.045	3.023
Duff depth (cm)	73.0	137.0	0.016	5.851
Graminoid cover (%)	144.5	65.5	0.001	9.344
Shrub cover (%)	129.0	81.0	0.026	3.947
Shrub height (in)	130.5	79.5	0.025	3.966
Shrub frequency	127.5	82.5	0.042	3.125

Sum of scores by condition, one-sided probability < Z and chi-square approximation with 1 df (see also Figure 5).

Figure 4. At Rachel Carson National Wildlife Refuge, Kittery, Maine, on an old field regenerating to eastern white pine and white spruce, nonnative Japanese barberry presents a substantial live fuel that is significantly more abundant than the shrub layer in a nearby uninvaded stand.

gravity at 12% moisture. Values for s were largely unavailable for shrubs, so we estimated the volume of dried (as opposed to 12% moisture) shrub twigs individually by measuring diameter, calculating the area of the cross section (at the middle of the twig), and then multiplying by the length and calculating density in grams per cubic centimeter. We averaged d^2 and s for each species over many sites.

Because we encountered almost no slash in most plots, we did not use a correction factor for slash. For calculations of hour fuels, we used actual counts on planar intercepts to derive percentages of species represented in each hour fuel class and used these percentages to weight d^2 and s. Species counts were on 10 planar intercepts per condition per site for 1-, 10- and 100-hour fuels, for a subset of sites. We also used percentages of each species found in the biomass plots to represent species by hour class along the planar intercepts. We checked the difference in the proportion of species based on the length of twigs found in the biomass plots with the number of twigs for each species on a few randomly chosen biomass plots and found that they differed <5%. Species were counted in only two of the five biomass plots per condition that we collected at each site. To further triangulate the data and because mostly 1- and 10-hour fuels populated the intercept data, we used the percent of each species recorded in the prism data to weight the constants for the 100-hour fuels.

We compared results from invaded stands with those from nearby stands that had few or no invasive plants. Variables included hour fuels by class in metric tons per hectare (tons per acre); biomass of nonwoody litter in metric tons per hectare (tons per acre); duff depth in centimeters (inches); fuel depth in meters (feet); percent cover of graminoids, forbs, shrubs, trees, and slash; shrub height (feet); and shrub frequency.

Means were prepared for each variable within each plot across all study areas. The data were not distributed normally; therefore, to distinguish significant differences (alpha = 0.05) between fuels in invaded and uninvaded conditions by forest type, we used a nonparametric test, the Wilcoxon two-sample test (PROC NPAR1WAY, SAS version 8.2; SAS Institute 1999), which uses rank scores to compare values that differ between the two groups. For tied ranks, the average of the number of tied values was used.

Results and Summary

In some of the uninvaded stands, we found patches of nonnative, invasive plants, but their stature and density were low enough that they rarely appeared in the uninvaded fuels data. There were wide discrepancies between invaded and uninvaded conditions in fuel attributes within some forest types. Visual aspects due to invasive plant presence were plain (i.e., Japanese barberry was in some locales a wall of thorny vegetation; Figure 4); therefore, the lack of true controls did not affect our ability to answer the original question of whether the invasive plants affect fuel loads.

We present only those results where significant differences were found between means for at least two forest types; thus, 100-hour fuels were not included in Table 2 or Figure 5. Within forest types, we did find some additional differences between study areas and will present results elsewhere that are not pertinent to the purpose of this article.

We found numerous differences between invaded and uninvaded fuels but none of these held across all four forest types (Table 2, Figure 5). In invaded conditions for the three forest types other than pitch pine, duff depth was lower (and often was accompanied by evidence of earthworm activity, which we did not measure), whereas shrub cover and shrub frequency were higher (Table 2). Shrub height was greater in invaded conditions within hardwoods, softwoods (e.g., Figure 6) and pitch pine but not in the mixed woods. Pitch pine and hardwoods had less nonwoody litter in the in-

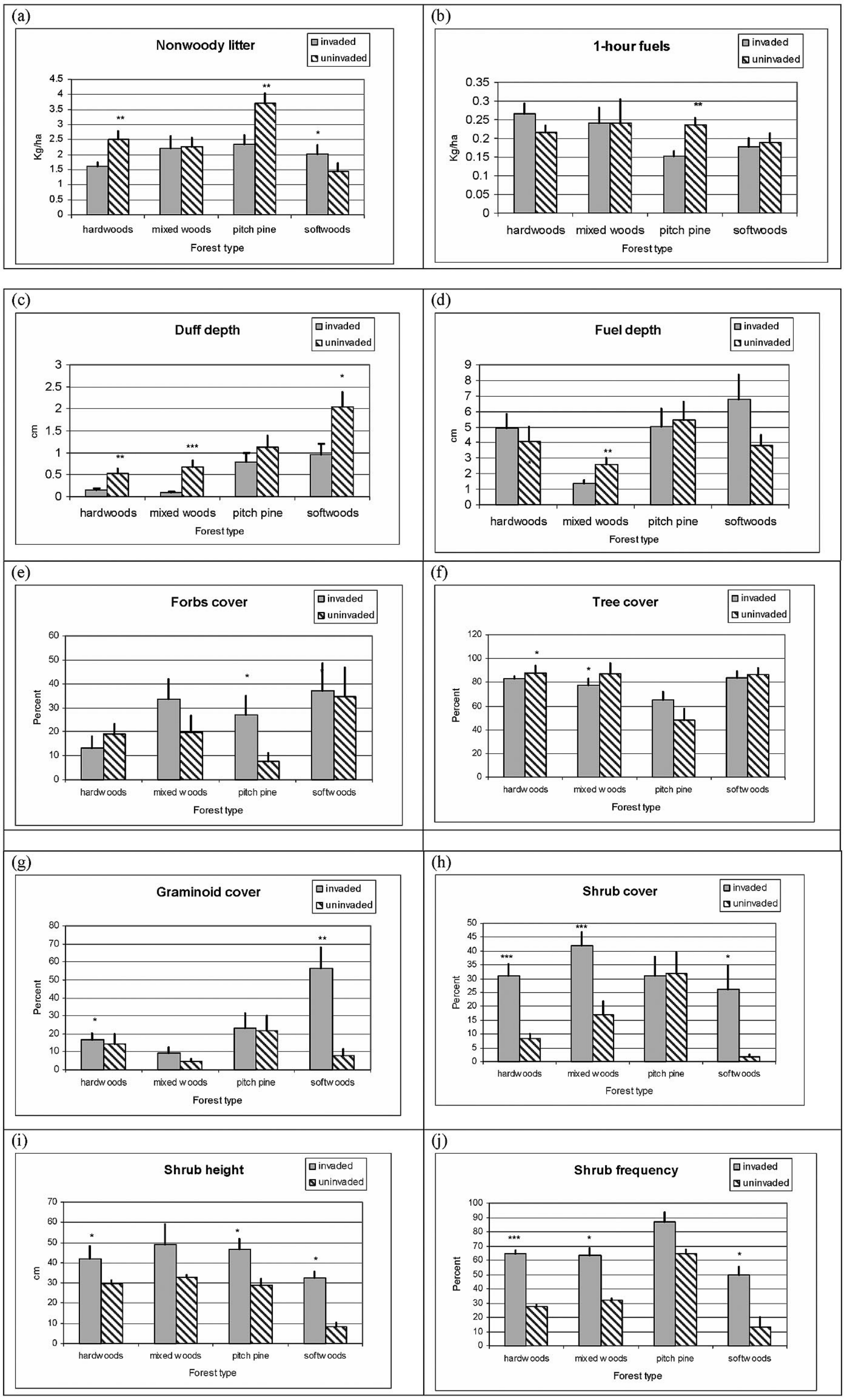

Figure 5. Comparison of fuels in invaded versus uninvaded forest stands in four broad forest type categories (Table 1). Whether invasive plants led to an increase or decrease in fuels, no one variable was consistent across all forest types. Highlights include (a) nonwoody litter (kilograms per hectare); (b) 1-hour fuels (kilograms per hectare); (c) duff depth (centimeters); (d) fuel depth (centimeters); and percent cover for (e) forbs, (f) trees, (g) graminoids, and (h) shrubs, plus (i) shrub height, and (j) shrub frequency.

vaded conditions whereas for softwoods this was higher. In hardwoods and mixed woods, tree cover was lower in invaded conditions but did not differ in softwoods or pitch pine. Graminoid cover was greater in invaded conditions in hardwoods and softwoods, and cover of forbs was greater in invaded pitch pine stands. Not shown in Figure 5, slash was higher in invaded conditions in the mixed wood stands; 10-hour fuels were lower in invaded conditions of pitch pine; and basal area was lower in invaded hardwoods but higher in invaded pitch pine (Table 2).

Our original emphasis in this study was woody plants, and there is some circularity in our method in that by definition, invaded stands often contained nonnative shrubs or trees that influenced the woody fuels. The uninvaded stands that we documented typically had a sparse shrub layer, with native shrubs such as low sweet blueberry or nannyberry (*Viburnum nudum* ssp. *cassinoides* [L.] Torr. & Gray), and in pitch pine, we often saw bearberry (*Arctostaphylos uva-ursi* [L.] Spreng.). We found that the invaded stands typically had abundant populations of nonnative shrubs that contributed to increased shrub height, shrub frequency, and percent cover of shrubs. Shrubs often included Asian honeysuckle, multiflora rose, or Japanese barberry, sometimes in a dense thicket (Figure 4). In dry conditions, shrubs could contribute a significant biomass of live fuels or, depending on the species involved, they might dampen fire effects by their shade, green foliage, and influence on litter volume, duff, and soil moisture conditions. Fuel models might not be sensitive enough to indicate the difference. These nonnative fuels could allow wildfire to spread rapidly by acting as ladder fuels and might increase the potential for crowning. We speculate that the resulting blaze could be larger and more intense than if the invasive shrubs were absent, but this probably will depend on seasonality and the shrub species present.

An unexpected result of this study was the documentation of nonnative, invasive grasses in some forest stands, including wood bluegrass, Japanese stiltgrass, fine-leaved sheep fescue (e.g., Figure 7), and on Cape Cod, sweet vernal grass. We found that such grasses formed a large patch of dense, continuous fine fuels that probably brown up differently than the native graminoids, which were present in small, sparse patches except in a few plots. Because these grasses significantly alter the phenology and continuity of fine fuels, we suggest that during an extreme drought in autumn, a wildfire could spread more easily than in a stand where such grasses are absent. Grass fuels might be considered hazard fuels and could require prioritization for control. Site-specific rather than general management recommendations are appropriate given that the species

A B

Figure 6. Fuels differed between invaded and uninvaded conditions at a former estate on Holbrook Island Sanctuary, Brooksville, Maine. (a) An uninvaded red spruce-balsam fir stand, adjacent to (b) the abandoned house site now occupied by wood bluegrass and Norway maple.

of invasive plants that occupy a forest in a given locale and the treatments allowable will determine which action might best achieve a goal of restoring native vegetation. A method of controlling invasive vegetation that is effective in one region or forest type might not be applicable or successful elsewhere; e.g., herbicides often are recommended for controlling some invasive plants but not all situations are conducive to their use. We found much information about successful control practices, although anecdotal, that is relevant to the Northeast on a list serve maintained by the Mid-Atlantic Exotic Pest Plant Council.

Implications for Similar Types of Studies. We compared fuels with and (mostly) without invasive plants. Although the comparison area method was not perfect, it was effective in the absence of longstanding studies that might have allowed for remeasurement of stands from a time before they became invaded. We suggest that if managers wish to compare treated and comparison areas, they will want to examine differences between the two areas that do not stem from the treatment itself. If these differences can be measured, they might be used as a covariate in a model (Eric Zenner, University of Minnesota, 2004, personal communication). Otherwise, it is advised to assume that only demonstrably significant differences can override the inadequate match up in conditions between treated and untreated areas, as we showed in the invasive fuel characterization study.

Figure 7. A surprising result of this study was documentation of extensive patches of nonnative invasive grasses in forests. Here, fine-leaved sheep fescue and wood bluegrass form a continuous fine fuel under bigtooth aspen on a 1947 burn at Acadia National Park, Bar Harbor, Maine.

In study areas where invasive plants have been established, their potential impact on fuels, biodiversity, and forest regeneration suggests that no invaded, untreated comparison area ought to be left unattended for long, regardless of the invasive plant species of interest or the study design selected. Although an untreated, invaded area would be useful for judging the efficacy of a treatment, seeds produced on invasive plants there might spread back into the treated area or to distant parts of the forest. In such a situation, the overall goal of controlling invasive plants and the hazard fuels they might present has a higher priority than maintaining an untreated area against which to measure response. The question of properly controlled experiments is second to overriding management priorities, such as reducing populations of invasive plants in the forest. As with any invasive plant problem, the best window in which to treat it is when the population is still small in the number of individuals and areal size. Common sense, resolute persistence, and an eye toward long-term results will help achieve a goal of restoring native forests, even if our picture of Northeastern forests 500 years ago remains a matter of lively discussion.

Literature Cited

ANDERSON, D.R., K.P. BURNHAMN, AND W.L. THOMPSON. 2000. Null hypothesis testing: Problems, prevalence, and an alternative. *J. Wildl. Manage.* 64(4):912–923.

ASBJORNSEN, H.L., L.A. BRUDVIG, C.M. MABRY, C.W. EVANS, AND H.M. KARNITZ. 2005. Defining reference information for restoring ecologically rare tallgrass oak savannas in the midwestern United States. *J. For.* 103(7): 345–350.

BENSON, A.B. (ED.). 1964. *Peter Kalm's travels in North America*. The English version of 1770 [revised from the original Swedish]. Dover Publications, New York. 797 p.

BETTS, M., AND J. LOO. 2002. A comparison of pre-European settlement forest characterization methodologies. *For. Chron.* 78(3):422–432.

BROWN, J.K. 1974. *Handbook for inventorying down woody material.* USDA For. Serv. Gen. Tech. Rep. INT-16, Intermountain Forest and Range Experiment Station, Ogden, UT. 24 p.

BROWN, J.K., R.D. OBERHEU, AND C.M. JOHNSON. 1982. *Handbook for inventorying surface fuels and biomass in the Interior West.* USDA For. Serv. Gen. Tech. Rep. INT-129, Intermountain Forest and Range Experiment Station, Ogden, UT. 48 p.

CHOJNACKY, D.C., R.A. MICKLER, L.S. HEATH, AND C.W. WOODALL. 2004. Estimates of down woody materials in eastern U.S. forests. *Environ. Manage*. 33(Suppl. 1):S44–S55.

CRONON, W. 1983. *Changes in the land: Indians, colonists, and the ecology of New England*. Hill and Wang, New York. 241 p.

DEHAYES, D.H., G.L. JACOBSON JR., P.G. SCHABERG, B. BONGARTEN, L. IVERSON, AND A.C. DIEFFENBACKER-KRALL. 2000. Forest responses to changing climate: Lessons from the past and uncertainty for the future. P. 495–540 in *Responses of northern U.S. forests to environmental change*, Mickler, R.A, R.A. Birdsey, and J. Hom, (eds.). Springer-Verlag, New York, NY.

DIBBLE, A.C. Northeast Chapter 5 in *Fire and nonnative invasive plants, rainbow series*, Vol. 6, Kapler Smith, J., K. Zouhar, and M. Brooks (eds.). USDA For. Serv., Missoula, MT (in press).

DIBBLE, A.C., J.C. BRISSETTE, AND M.L. HUNTER JR. 1999. Putting community data to work: Some understory plants indicate regeneration habitat for red spruce. *For. Ecol. Manage*. 114: 275–291.

Dibble, A.C., C.A. Rees, M.J. Ducey, and W.A. Patterson III. 2003a. Fuel bed characteristics of invaded forest stands. P. 26–29 in *Using fire to control invasive plants: What's new, what works in the Northeast: 2003 workshop proceedings*. University of New Hampshire Cooperative Extension, Durham, NH. Available online at www.ceinfo.unh.edu/Pubs/ForPubs/WPUFCI03.pdf; last accessed Oct. 21, 2005.

Dibble, A.C., W.A. Patterson III, and R.H. White. 2003b. Relative flammability of native and invasive exotic plants of the Northeastern U.S. P. 34–37 in *Using fire to control invasive plants: What's new, what works in the Northeast: 2003 workshop proceedings*. University of New Hampshire Cooperative Extension, Durham, NH. Available online at www.ceinfo.unh.edu/Pubs/ForPubs/WPUFCI03.pdf; last accessed Oct. 21, 2005.

Dibble, A.C., C. A. Rees, P.E. Sendak, and J.C. Brissette. 2004. *Vegetation of forested uplands in the Massabesic Experimental Forest*. USDA For. Serv. Gen. Tech. Report NE-320, Northeastern Research Station, Newtown Square, PA. 71 p.

Fernald, M.L. 1950. *Gray's manual of botany*, 8th ed. American Book Co., New York. 1632 p.

Ford, E.D. 2000. *Scientific method for ecological research*. Cambridge University Press, Cambridge, UK. 564 p.

Forest Products Laboratory. 1955. *Wood handbook: Basic information on wood as a material of construction with data for its use in design and specification.* Agricultural handbook no. 72, USDA For. Serv., Madison, WI, US Government Printing Office, Washington, DC. 528 p.

Frelich, L.E., M.W. Cornett, and M.A. White. 2005. Controls and reference conditions in forestry. The role of old-growth and retrospective studies. *J. For*. 103(7): 339–344.

Goebel, C., T.C. Wyse, and R.G. Corace III. 2005. Determining reference ecosystem conditions for disturbed landscapes within the context of contemporary resource management issues. *J. For*. 103(7):351–356.

Green, R.H. 1979. *Sampling design and statistical methods for environmental biologists.* John Wiley and Sons, New York. 257 p.

International Panel on Climate Change. 2001. *Climate change 2001: The scientific basis,* Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, and D. Xiaosu (eds.). University Press, Cambridge, UK. Available online at www.grida.no/climate/ipcc_tar/wg2/554.htm; last accessed Oct. 21, 2005.

Kenefic, L.S., A.S. White, A.R. Cutko, and S. Fraver. 2005. Reference stands for silvicultural research: A Maine perspective. *J. For.* 103(7):363–367.

Keppie, D.M. 1990. To improve graduate student research in wildlife education. *Wildl. Soc. Bull.* 18:453–458.

LANDFIRE. Available online at www.landfire.gov; last accessed Oct. 21, 2005.

Lawrence, B. 1991. The early American wilderness as the explorers saw it. Paragon House, New York. 245 p.

Lindholdt, P.J. (ed.). 1988. *John Josselyn, colonial traveler: A critical edition of two voyages to New-England.* University Press of New England, London. 221 p.

McCune, B., and J.B. Grace. 2002. *Analysis of ecological communities.* MJM Software Design, Gleneden Beach, OR. 300 p.

Mid-Atlantic Exotic Pest Plant Council. Available at www.ma-eppc.org; last accessed Oct. 21, 2005.

Pielou, E.C. 1991. *After the Ice Age: The return of life to glaciated North America*. University of Chicago Press, Chicago, IL. 366 p.

Richburg, J.A., A.C. Dibble, and W.A. Patterson III. 2001. Woody invasive species and their role in altering fire regimes of the Northeast and Mid-Atlantic states. P. 104–111 in *Proc. of the invasive species workshop: The role of fire in the control and spread of invasive species*, Galley, K.E.M., and T. P. Wilson, (eds.). Fire conf. 2000: The 1st National Congress on fire ecology, prevention, and management, misc. publ. 11, Tall Timbers Research Station, Tallahassee, FL.

SAS Institute. 1999. *The SAS System: SAS OnlineDoc*, version 8 [Base SAS Software, SAS/GRAPH Software: Reference]. SAS Institute, Inc., Cary, NC.

Stephens, S.L., and P.Z. Fule. 2005. Western pine forests with continuing frequent fire regimes: Possible reference sites for management. *J. For*. 103(7):357–362.

Vernooy, R. and C. McDougall. 2003. Principles for good practice in participatory research: Reflecting on lessons from the field, Chap. 6 in *Managing natural resources for sustainable livelihoods: Uniting science and participation*, Pound, B., S. Snapp, C. McDougall, and A. Braun (eds.). Earthscan/The International Development Research Centre, Ottawa, Canada (IDRC), ISBN 1-55250-071-3. 260 p. Available online at www.idrc.ca/en/ev-43440-201-1-D0_TOPICS.html; last accessed Oct. 21, 2005.

Whitney, G.G. 1994. *From coastal wilderness to fruited plain: A history of environmental change in temperate North America 1500 to the present*. Cambridge University Press, Cambridge, UK. 451 p.

Williams, M. 1992. *Americans and their forests: A historical geography*. Cambridge University Press, Cambridge, UK. 599 p.

Wilson, J.S. 2005. Nineteenth century lumber surveys for Bangor, Maine: Implications for pre-European settlement forest characteristics in northern and eastern Maine, U.S.A. *J. For.* 103(5):218–223.

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